

Field-scale monitoring of the long-term impact and sustainability of drainage water reuse on the west side of California's San Joaquin Valley

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Diminishing freshwater resources have brought attention to the reuse of degraded water as a water resource rather than a disposal problem. Drainage water from tile-drained, irrigated agricultural land is degraded water that is often in large supply, but the long-term impact and sustainability of its reuse on soil is unknown. Similarly, nothing is known of the ramifications of terminating drainage water reuse. The objective of this study is (i) to monitor the long-term impact on soil chemical properties and thereby the sustainability of drainage water reuse on a marginally productive, saline-sodic, 32.4 ha field located on the west side of California's productive San Joaquin Valley and (ii) to assess spatially what happens to soil when drainage water reuse is terminated. The monitoring and assessment were based on spatial chemical data for soil collected during 10 years of irrigation with drainage water followed by 2 years of no applied irrigation water (only rainfall). Geo-referenced measurements of apparent soil electrical conductivity (EC_a) were used to direct the soil sampling design to characterize spatial variability of impacted soil properties. Chemical analyses of soil samples were used (i) to characterize the spatial variability of salinity, Na, B, and Mo, which were previously identified as critical to the yield and quality of Bermuda grass (*Cynodon dactylon* (L.) Pers.) grown for livestock consumption and (ii) to monitor their change during the 12 year study. Soil samples were taken at 0.3 m increments to a depth of 1.2 m at each of 40 sample sites on five occasions: August 1999, April 2002, November 2004, August 2009, and May 2011. Drainage water varying in salinity ($1.8\text{--}16.3\text{ dS m}^{-1}$), SAR ($5.2\text{--}52.4$), Mo ($80\text{--}400\text{ }\mu\text{g L}^{-1}$), and B ($0.4\text{--}15.1\text{ mg L}^{-1}$) was applied from July 2000 to June 2009. Results indicate that salts, Na, Mo, and B were leached from the root zone causing a significant improvement in soil quality from 1999 to 2009. Salinity and SAR returned to original levels or higher in less than two years after termination of irrigation. Boron and Mo showed significant increases. Long-term sustainability of drainage water reuse was supported by the results, but once application of irrigation water was terminated, the field quickly returned to its original saline-sodic condition.

1. Introduction

Water scarcity is a growing concern throughout the world, particularly in arid and semi-arid regions where irrigated agriculture is either an essential element of the economy or crucial to the survival of the populace, including water-vulnerable areas like China, Pakistan, India, northwestern Mexico, Middle East,

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Environmental impact

Water vulnerable arid and semi-arid agricultural regions (*e.g.*, northeast China, Middle East, north and eastern Africa, southwestern USA, India, and Pakistan) are seriously considering and using degraded waters as a major potential supplemental source of irrigation water. Yet, nothing is known of the long-term impact or sustainability. This research (1) confirms the long-term sustainability of drainage water reuse on marginally productive saline-sodic soil, (2) demonstrates advanced information technology for assessing field-scale impacts of degraded water reuse, and (3) indicates what can happen when drainage water reuse on saline-sodic soil is terminated. The relevance and impact of this research is geographically broad since it pertains to the viability of degraded water reuse on irrigated agricultural lands throughout the world.

northern and eastern Africa, portions of eastern Australia, and southwestern USA. Approximately, one-fifth of the world's population live in areas of physical scarcity of water (FAO, 2007).¹ In addition, there are localized areas of highly productive irrigated agriculture, such as California's San Joaquin, Imperial, and Coachella Valleys that experience serious water shortages due to intermittent drought conditions, which deleteriously impact productivity. Climate change forecasts indicate that extremes in wetness and dryness can be expected in the future on a more regular basis, with arid zones such as the southwestern USA experiencing extended periods of drought and areas such as North Dakota and Minnesota's Red River Valley continuing to receive above normal precipitation (IPCC, 2007).² Whether viewed from a global, regional, or local level, water scarcity is recognized as an issue of current and future concern.

Diminishing freshwater resources due to increased urban demands, increased incidence of drought, and continued degradation from point and non-point sources of pollution have brought attention to the reuse of degraded water as a potential water resource rather than as a disposal problem, particularly in water scarce regions of the world. Degraded waters are a means of supplementing diminishing water resources. Degraded waters include agricultural drainage water, municipal wastewater, CAFO (confined animal feeding operations) wastewater, food processing wastewater, urban and agricultural runoff, and industrial wastewater.

Worldwide, irrigated agriculture accounts for 70% of the freshwater withdrawn for human use (World Resources Institute, 2000).³ Ironically, some of the world's most productive irrigated agricultural regions lie in areas of potential water vulnerability, such as California's San Joaquin Valley. It is inevitable that agriculturally important areas relying on vulnerable water resources will be drawn to the reuse of degraded waters. The reuse of degraded water as a supplemental source of water for irrigated agriculture throughout the world is particularly appropriate for areas that are pushing their water resources to the limit and are susceptible to drought and/or increasing urban demands for water. The most readily available degraded water for reuse on irrigated agriculture is drainage water. Yet, little is known of the long-term impact of drainage water on soil properties and whether or not it is sustainable nor is anything known of what will happen once the reuse of drainage water is terminated. The only information is what can be inferred from the detrimental impacts found in drainage water storage reservoirs, such as Kesterson Reservoir. Kesterson Reservoir served as the storage site for drainage waters from the west side of the San Joaquin Valley. Toxic levels of Se in drainage water caused birth defects in water fowl, which led to the closure of Kesterson Reservoir in 1987.

Irrigation and drainage are closely associated and are usually interdependent regardless of geographic location. The west side of the San Joaquin Valley (WSJV) exemplifies the close association of irrigation and drainage. The WSJV is an agriculturally productive area that suffers from a concern about future ample sources of water for irrigation and what to do with the drainage water associated with irrigation. The WSJV drainage waters are known to have deleterious effects on plants, grazing animals, and wildlife due to the salts and trace elements (*e.g.*, As, B, Mo, and Se) that they contain (Letey, 1986; Shannon, 1997).^{4,5} As

a consequence of the closing of the Kesterson Reservoir as an drainage water outlet for the WSJV, drainage water has become a serious disposal problem. Currently, no comprehensive plan exists for the disposal of drainage water for the WSJV. Without a means of disposing of drainage water, water tables will inevitably rise too high to maintain productivity and WSJV farms will not be sustainable. Evaporation ponds are currently used to dispose of the drainage water, but this requires 1 ha of evaporation pond for every 10 ha of tile-drained, irrigated agricultural land; consequently, considerable land is taken out of productivity. Significantly reducing drainage volumes would reduce the land needed for evaporation ponds and bring land back into productivity.

Arid zone agricultural lands throughout the world, exemplified by the WSJV, typically suffer from diminishing water resources, drainage disposal problems, and sodic and/or salt-affected soils that reduce crop yields. One means of reducing drainage water volumes, reusing drainage water as an alternative water supply, and returning non-productive soil to productivity is the reuse of drainage water on marginally productive, saline-sodic soils. Saline-sodic soils are susceptible to infiltration and permeability problems when low electrical conductivity (EC) water is applied to leach out the salts. Rather, sufficient salinity is needed in the irrigation water to maintain infiltration and permeability, making drainage water a potentially viable irrigation water source and means of reclamation when applied to a saline-sodic field with a salt-tolerant crop. The growth of a salt-tolerant crop during drainage water reuse is important as a source of revenue and as a means of maintaining soil structure and water flow properties through the aggregating influence on soil by the root system. It is hypothesized that the reuse of drainage water on saline-sodic soils will reduce drainage volumes, thereby reducing the land needed for evaporation ponds, and will reclaim these soils by maintaining permeability and leaching excess Na and salts, bringing non-productive land back into production by supporting a revenue-generating salt-tolerant forage crop. The viability of this approach has been evaluated over the short term by Corwin *et al.* (2003, 2008).^{6,7} But, the more critical issue of long-term sustainability of drainage water reuse is a gap in our knowledge (Corwin and Bradford, 2008).⁸

There are numerous papers that have dealt with drainage water reuse (Westcot, 1988; Rhoades, 1989; Grattan and Rhoades, 1990; Ayars *et al.*, 1993; Tanji and Karajeh, 1993; Willardson *et al.*, 1997; Goyal *et al.*, 1991; Mitchell *et al.*, 2000; Oster and Grattan, 2002; Grattan and Oster, 2003; Grattan *et al.*, 2004, and Grieve *et al.*, 2004).⁹⁻²⁰ but none have looked at impact and sustainability from a field-scale perspective over the long term. Furthermore, no studies have evaluated what happens to soils once degraded water reuse has terminated. Does it return to its original state and if so, how quickly? Currently, the short-term study (*i.e.*, 5 years) by Corwin *et al.* (2008)⁷ is the only field-scale evaluation of spatio-temporal impacts of drainage water reuse on soil chemical properties.

Little field-scale documentation currently exists on long-term (*i.e.*, 10 years or longer) sustainability of degraded water reuse, in particular a knowledge gap exists for the reuse of drainage water (Corwin and Bradford, 2008).⁸ The preliminary drainage water reuse work by Corwin *et al.* (2008)⁷ indicated that a long-term

(i.e., 10 year) evaluation of the sustainability of drainage water reuse in California's WSJV is needed due to concerns with Mo and Se accumulation and increases in soil pH. It is the objective of this study (i) to monitor spatially the long-term impact of drainage water reuse on the chemical properties of a marginally productive, saline-sodic field located in the WSJV, (ii) to evaluate its sustainability in light of the fact that drainage water from the WSJV contains salts and trace elements (e.g., B, Mo, Se) that could detrimentally affect soil quality, and (iii) to monitor and evaluate the consequence of terminating the reuse of drainage water.

2. Methods and materials

Assessments of spatio-temporal change in soil quality due to drainage water reuse were made by periodically characterizing the spatial variability of soil chemical properties influencing the intended use of the soil, which in this case was to grow Bermuda grass (*Cynodon dactylon* (L.) Pers.) as forage for livestock. A forage crop was planted at the study site since it provided revenue from livestock while requiring minimal input of labor and resources. Bermuda grass was selected as the forage crop because it is salt tolerant with a threshold salinity of 6.9 dS m^{-1} (Maas and Hoffman, 1977).²¹ Spatial variability of soil chemical (and some physical) properties was determined using the apparent soil electrical conductivity (EC_a) directed soil sampling approach developed by Corwin and Lesch (2003, 2005a, 2005b).^{22–24} Sustainability was evaluated based on the impact of drainage water reuse on soil quality with respect to the soil's intended use of supporting forage growth for livestock.

Study site

A 10-year drainage water reuse study was initiated in August 1999 on a 32.4 ha saline-sodic field (latitude $36^\circ 11' 24.827'' \text{ N}$, longitude $119^\circ 52' 45.455'' \text{ W}$) located on Westlake Farm, which resides in WSJV's Kings County. The soil is part of the Lethent clay loam series and is classified as a fine, montmorillonitic, thermic, Typic Natrargid (USDA, 1986).²⁵ Further details of the site preparation can be found in Kaffka *et al.* (2002)²⁶ and Corwin *et al.* (2003).²⁷ Fig. 1 provides an aerial view of the 32.4 ha field and a map of its location within CA.

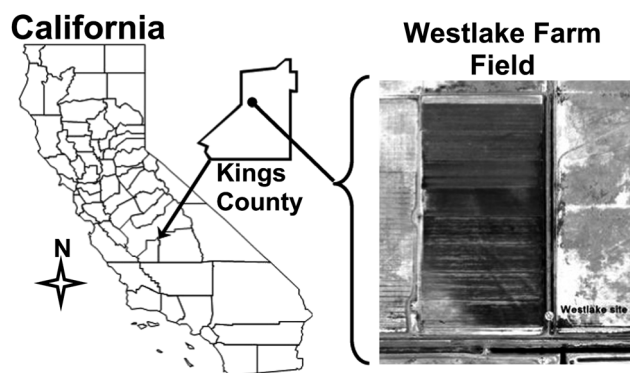


Fig. 1 Map showing location of Westlake Farm 32.4 ha study site on the west side of California's San Joaquin Valley.

The study site was selected to represent a worst case scenario and the greatest possible challenge for testing and evaluating the reuse of drainage water on marginally productive saline-sodic soil. The site was extremely sodic with initial SAR ranging from 30.0 to 88.8 through the root zone (i.e., 0–1.2 m) and extremely saline with initial EC_e ranging from 12.8 to 36.6 dS m^{-1} . The field had been left fallow by the farm owner since 1992 due to its low productivity. The infiltration rate and permeability at the southern end of the field were so slow that leaching was impractical with the available high quality irrigation water from the California Aqueduct. In essence, the field had no value to the farmer and was regarded as commercially irreparable.

Apparent soil electrical conductivity survey

Four EC_a surveys using mobile electromagnetic induction (EMI) equipment were conducted over the 12 years of the drainage water reuse project: August 1999, April 2002, November 2004, and April/May 2011. The initial EC_a survey, which was conducted 12–16 Aug. 1999, consisted of a grid of EC_a measurements arranged in a 32 (row) \times 12 (position within row) pattern for a total of 384 sites across the study site. All 384 sites were geo-referenced using a Trimble Pro-XRS GPS system† (Trimble, Sunnyvale, CA, USA) with sub-metre precision. Electromagnetic induction EC_a measurements were taken with a Geonics single-dipole EM38 Electrical Conductivity Meter† (Geonics Ltd., Mississauga, Ontario, Canada). Measurements were taken in the horizontal (EM_h) and vertical (EM_v) dipole modes to provide shallow (0–0.75 m) and deep (0–1.5 m) measurements of EC_a , respectively. Measurements were taken at locations approximately 20–30 m apart.

Surveys conducted after 1999 were more intensive. Upgrades in our EMI and GPS equipment permitted the continuous and automatic measurement of EC_a in both horizontal and vertical dipole modes every 3 s rather than discrete measurements from one location to the next. This resulted in traverses spaced roughly 8 m apart and 4 m between each location within a traverse.

The second EC_a survey was conducted from 8–12 Apr. 2002 using the upgraded mobile EMI equipment and following the EC_a survey methodology that eventually led to the protocols paper by Corwin and Lesch (2005a).²³ Details of this mobile EMI equipment can be found in Corwin and Lesch (2005a, 2005b).^{23,24} The survey consisted of EC_a measurements (i.e., both EM_h and EM_v) taken at 22 177 locations within the field. An associated GPS reading was taken with each set of EM_h and EM_v measurements. The third and fourth EC_a surveys were conducted on 14–15 Dec. 2004 and 19–22 Apr. 2011, respectively. These surveys followed the same protocols used in the April 2002 survey. The field was at field capacity when all EC_a surveys were conducted to make certain that they were at the same water content for comparison purposes. Fig. 2 shows the EM_h and EM_v maps for the 2002, 2004, and 2011 surveys. For a figure of the 1999 EC_a survey the reader is referred to the paper by Corwin *et al.* (2003).²⁷

† The citation of particular products or companies is for the convenience of the reader and does not imply any endorsement, guarantee, or preferential treatment by the U.S. Department of Agriculture.

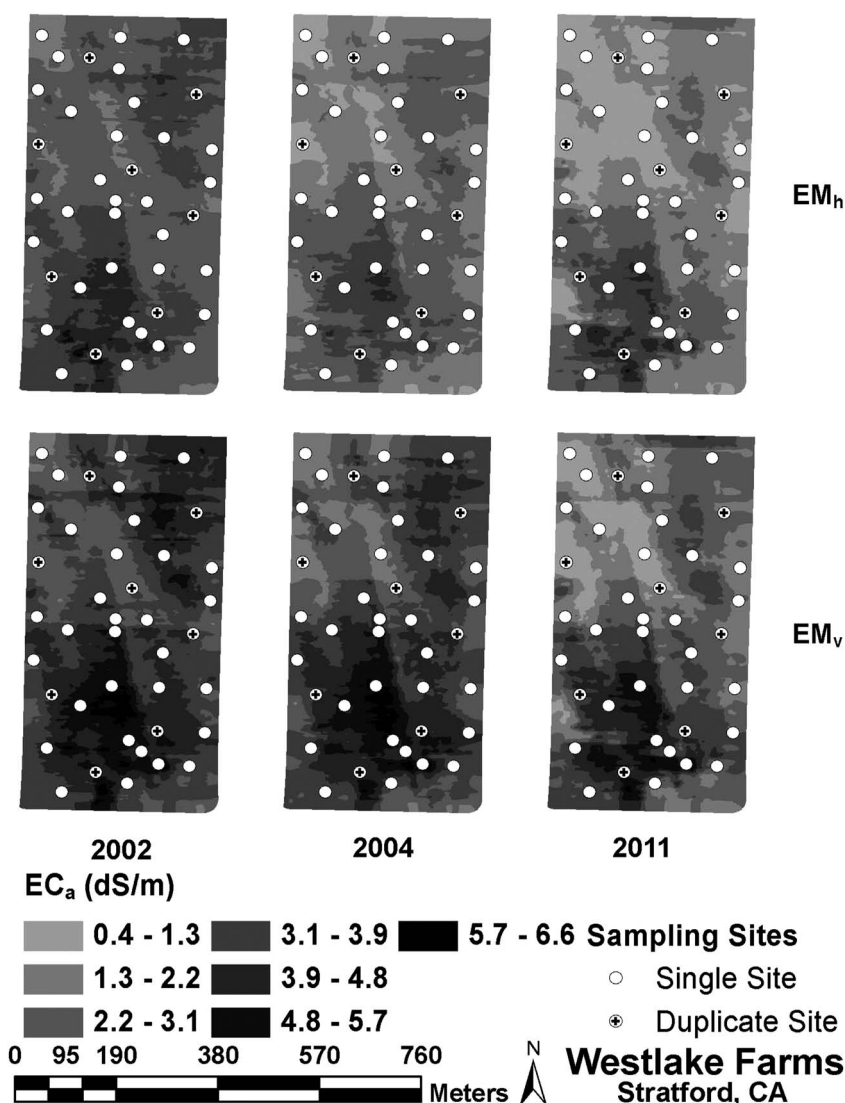


Fig. 2 Spatial comparison of the 2002, 2004, and 2011 electromagnetic induction EC_a surveys of EM_h and EM_v measurements and the 40 soil sampling sites (circles). Circles with crosshairs indicate sites where duplicate samples were taken for local-scale variation analysis discussed in Corwin *et al.* (2003).⁶ EC_a , apparent soil electrical conductivity ($dS\ m^{-1}$); EM_h , electromagnetic induction measured in the horizontal dipole mode; EM_v , electromagnetic induction measured in the vertical dipole mode. Years 2002 and 2004 are taken from Corwin *et al.* (2008).⁷

Visual variations in the patterns of the geospatial EC_a measurements for the three EC_a surveys were used to indicate spatial changes in dynamic properties, such as salinity, which influence the EC_a measurement. In general, visual variations in spatial patterns were minor, indicating spatial stability over time, which is well documented (Farahani and Buchleiter, 2004; Corwin *et al.*, 2006).^{28,29} Even though the magnitude of EC_a measurements changed over time, the visual patterns were relatively stable, which is a reflection of the texture and its influence on water flow and solute distributions.

Statistical sampling methodology

Soil core sample sites were selected using geo-referenced EC_a survey data as a surrogate for the spatial variation of soil properties known to influence EC_a including soil texture, water content, salinity, organic matter, cation exchange capacity, bulk density, and organic matter. The general sampling approach is discussed in

Lesch *et al.* (1995)³⁰ and Lesch (2005)³¹ and details of the sampling approach for Westlake Farm are found in Corwin *et al.* (2003).²⁷

Using the EC_a data from the 1999 EMI survey and ESAP software (Lesch *et al.*, 2000),³² 40 soil sample sites were selected that characterized the spatial variability in EC_a . The 40 sites were chosen (i) to represent about 95% of the observed range in the bivariate EMI survey data, (ii) to represent the average of the EC_a readings for the entire field, and (iii) to be spatially distributed across the field minimizing any clustering. The number of sites selected was based on the following criteria: (i) variability of the field as revealed by the EC_a survey, (ii) available resources to conduct soil sample analyses, and (iii) intended use of the data based on the objective of the study.

Soil core sampling

At each of the 40 sites, soil-core samples were taken at 2 points (*i.e.*, two sets of soil cores per site) roughly 5 centimetres apart.

Soil cores were taken at 0.3 m increments to a depth of 1.2 m. One set of soil cores was designated for soil chemical property analysis and the other set for soil physical property analysis. A total of 320 soil samples were taken (160 soil chemical property samples, 160 soil physical property samples). Fig. 2 shows the location of the 40 selected soil-core sites.

To observe temporal changes resulting from the application of drainage water, soil core samples were taken at the same 40 selected sample site locations at 5 different dates over the 12 years of the study: 19–23 Aug. 1999, 15–17 Apr. 2002, 30 Nov.–3 Dec. 2004, 18–20 Aug. 2009, and 2–6 May 2011. All soil cores were kept in refrigerated storage prior to air-drying and sieving (2 mm sieve), which occurred within a few days after their collection.

Soil physical and chemical analyses

The soil cores were analyzed for a range of physical and chemical properties considered important for the assessment of soil quality of an arid zone soil when the goal was the production of forage. The soil chemical properties included: electrical conductivity of the saturation extract (EC_e); pH_e ; anions (HCO_3^- , Cl^- , NO_3^- , SO_4^{2-}) and cations (Na^+ , K^+ , Ca^{2+} , Mg^{2+}) in the saturation extract; trace elements (B, Se, As, Mo) in the saturation extract; $CaCO_3$; gypsum; cation exchange capacity (CEC); exchangeable Na^+ , K^+ , Mg^{2+} and Ca^{2+} ; exchangeable sodium percentage (ESP); SAR; total C; and total N. The 192 soil samples designated for analysis of soil physical properties were analyzed for saturation percentage (SP; moisture content of the soil at saturation expressed as a percentage on a gravimetric basis), volumetric water content (θ_v), bulk density (ρ_b), and clay content.

The soil sample preparation, and chemical and physical methods used for each analysis were from ASA Agronomy Monograph No. 9 Parts 1 and 2 (Klute, 1986;³³ Page *et al.*, 1982;³⁴ respectively), except for total C and N, which were analyzed with a Leco C-N 2000 Analyzer¹ (Leco Corporation, St. Joseph, MI).

GIS and map preparation

A geographic information system (GIS) was used to display and manipulate the spatial data. All spatial data were entered into the GIS with the commercial GIS software ArcView 3.3. Interpolated maps of the soil chemical properties most significantly influencing soil quality were prepared using ordinary inverse-distance-weighting (IDW) interpolation. A comparison of IDW interpolation to kriging for all sampling times using jackknifing showed a general improvement in prediction of the interpolated values using IDW; consequently, all interpolations were done using IDW.

Irrigation with drainage water

Bermuda grass was established in the southern half of the field in the spring of 2000 and in the northern half in the summer of 2000. Once the Bermuda grass was established, the reuse of drainage water as the primary source of irrigation water began. On a few occasions the drainage water supply was depleted and irrigation with municipal waste water occurred, but this represented less than 4% of the water applied. The field was divided into 8 paddocks so livestock feeding on the Bermuda grass could be

placed in dry paddocks during irrigation. The field was irrigated with drainage water two paddocks at a time, which prevented livestock from compacting soil in those paddocks that were being irrigated.

Drainage water was first applied in July of 2000. Drainage water from an on-farm holding pond was applied to the field site usually from July to September from 2000–2009. The overall volume-weighted average EC_{dw} applied was 4.66 dS m^{-1} . The drainage water applied from 2000 to 2009 was 1.06, 0.73, 0.80, 0.93, 0.97, 0.96, 0.94, 1.09, 0.92 and 0.98 m, respectively, which averaged 0.94 m annually.

Annual rainfall from 2000–2009 was 0.09, 0.32, 0.74, 0.06, 0.11, 0.11, 0.14, 0.08, 0.17, and 0.10 m, respectively, which averaged 0.19 m year^{-1} . However, the average rainfall from 2000–2009 is misleading. Most years the rainfall was well below the average due to ongoing drought conditions in California's WSJV. During the year of heaviest rainfall, 2002, most of the rainfall occurred in a very short period of time and left the field as runoff. The rainfall that occurred in other years did not penetrate due to the high Na levels in the soil, which caused the rainfall to pond on the surface and evaporate. Chloride data revealed the overall leaching fraction (*i.e.*, LF = fraction of irrigation water leaving the root zone) for the entire field to be 0.16 over the 10 years from 2000–2009. Using estimates of the applied irrigation water, evapotranspiration and LF, the portion of water penetrating the soil due to rainfall was estimated to be less than 5%; consequently, rainfall from 2000–2009 was not a significant input of water.

Table 1 shows a detailed analysis of the chemical composition of the reused drainage water. The detailed chemical analysis was conducted on water samples taken from the irrigation delivery system once each year during the months of highest irrigation volumes (*i.e.*, June–August) from 2000–2009, except for 2007. Available resources only allowed for the detailed analysis of a single water sample each year. The range in chemical composition for those properties potentially influencing water quality varied, with EC ranging from 1.80 to 16.26 dS m^{-1} , SAR from 5.2 to 52.4, B from 0.4 to 15.1 mg L^{-1} , Se from <1 to $700 \text{ } \mu\text{g L}^{-1}$, and Mo from 80 to $400 \text{ } \mu\text{g L}^{-1}$. The drainage water composition tended to reflect the properties of the soil, irrigation management, and crop history of those lands in production that drained into the evaporation pond. The composition varied from one year to the next as land went into and out of production and crops varied.

Data analysis

A mixed linear ANOVA modeling technique was used to determine the significance of change by depth for shift in the field means over time of soil chemical properties, which corresponds to the *F*-test for no shift in the mean level over time ($t_1 - t_2 = 0$), and the significance of change of spatial variation, which corresponds to the *F*-test for no dynamic variation ($\Phi_{ts}^2 = 0$). This approach is described in Corwin *et al.* (2006).²⁹ Significance was assigned to probability levels $p \leq 0.05$.

3. Results and discussion

A detailed discussion of the initial conditions at the Westlake Farm study site is provided by Corwin *et al.* (2003)⁶ and

Table 1 Chemical composition of reused drainage water applied to Westlake Farm field site. Water samples were taken each year during the summer from 2000–2006 and 2008–2009^a

Chemical property	Sample date of reused drainage water											
	22 Aug. 2000	24 Aug. 2001	13 Aug. 2002	26 July 2003	29 Aug. 2004	27 July 2005	15. Aug. 2006	9 June 2008	25 June 2009			
EC _e (dS m ⁻¹)	4.33	16.26	4.65	5.08	1.80	5.01	5.86	4.49	5.88			
pH _e	7.4	7.9	7.4	7.4	7.7	8.5	8.7	8.4	8.8			
Na ⁺ (meq. L ⁻¹)	23.7	219.3	28.0	37.4	11.5	38.1	44.1	34.4	47.9			
K ⁺ (meq. L ⁻¹)	0.5	0.5	0.5	1.3	0.2	0.5	1.0	0.4	1.1			
Ca ²⁺ (meq. L ⁻¹)	25.3	21.9	24.3	25.6	6.3	7.3	7.5	7.1	7.7			
Mg ²⁺ (meq. L ⁻¹)	13.5	13.1	14.7	10.6	1.2	9.3	13.3	10.2	12.4			
SO ₄ ²⁻ (meq. L ⁻¹)	54.6	204.3	59.1	65.5	12.1	36.2	43.2	37.0	42.8			
Cl ⁻ (meq. L ⁻¹)	6.4	34.5	6.2	4.9	5.5	10.0	14.2	6.5	17.7			
HCO ₃ ⁻ (meq. L ⁻¹)	1.2	1.5	1.2	1.3	1.3	6.4	6.6	6.2	6.7			
NO ₃ ⁻ (meq. L ⁻¹)	<0.1	0.6	<0.1	<0.1	0.7	0.4	0.2	0.5	0.1			
PO ₄ ³⁻ (meq. L ⁻¹)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1			
SAR	5.4	52.4	6.3	8.8	5.9	13.2	13.7	5.2	15.1			
B (mg L ⁻¹)	1.3	15.1	1.6	2.0	0.4	1.4	1.8	0.7	6.3			
Se (μg L ⁻¹)	60	700	20	30	<1	<1	<1	<1	<1			
As (μg L ⁻¹)	<1	<1	<1	<1	<1	<1	<1	<1	<1			
Mo (μg L ⁻¹)	120	400	120	160	80	82	132	134	176			

^a EC_a = apparent soil electrical conductivity; SAR = sodium adsorption ratio.

a short-term assessment of the impact of drainage water reuse is provided by Corwin *et al.* (2008).⁷ Corwin *et al.* (2008)⁷ established that the chemical properties of salinity as measured by the EC of the saturation extract (EC_e), sodium adsorption ratio (SAR), pH, B, Mo, and Se were of greatest concern due to their potential influence on the yield and quality of Bermuda grass and on the general environmental health of the soil; consequently, attention is focused on these chemical properties in this paper.

Temporal trends of EC_e, SAR, B, Mo, Se, and pH_e during drainage water reuse (1999–2009)

Table 2 shows the correlations between EC_a measurements taken with EMI in 1999 and soil physical and chemical properties for the 5 soil sampling times of 1999, 2002, 2004, 2009, and 2011. Correlations that are significant ($p \leq 0.05$) indicate those soil properties that were spatially characterized with the RSSD approach using the 1999 EMI EC_a survey; as a result, the field means and spatial distributions of these significantly correlated properties can be considered reliable. Soil properties that are not significantly correlated with EC_a were not spatially characterized with the RSSD approach, which casts doubt on the accuracy of their field means. For all five sample times, salinity (*i.e.*, EC_e), SAR, and B are significantly correlated with EC_a, while Mo, Se, and pH_e are significantly correlated in almost every instance except in 2004 for Mo and Se and in 1999 for pH_e; consequently, it can be concluded that the sampling design generated from the EC_a survey in 1999 characterized the spatial distribution of EC_e, SAR, B, Mo, Se, and pH_e for each sample time and reliably determined their field means.

Tables 3–6 provide the basic statistics of the soil properties of interest for 1999, 2004, 2009, and 2011 at the depth increments of 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m. These statistics show the extent of change in the field means of soil properties over the 10-year period of drainage water reuse (*i.e.*, 1999–2009) followed by 2 years when the field was no longer irrigated and the forage crop was only rain fed (*i.e.*, 2009–2011). The general trend from 1999 to 2009 was a decrease in the field means at nearly every depth increment for EC_e, SAR, B, and Mo (Tables 3–6 and Fig. 3). The general downward trend of these chemical properties reflects the effect of leaching the soil profile with the application of drainage water having a SAR generally below 15, B generally less than 2 mg L⁻¹, Mo generally less than 200 μg L⁻¹, and a volume-weighted average salinity of 4.66 dS m⁻¹. The decrease in these properties is attributable to two effects resulting from the reuse of drainage water of high salinity: (i) improved infiltration and (ii) the leaching of solutes due to the application of water in higher volumes than consumptive water use by the crop.

From 1999 to 2009 the field mean of Se increased at all depth increments (see Tables 3–6). However, Se actually decreased from 2004 to 2009, reflecting the lower Se levels present in the reused drainage water for 2004–2009, which were consistently <1 μg L⁻¹ (see Table 1). The overall increase in Se from 1999–2009 was a consequence of higher levels of Se present in the reused drainage water from 1999–2004, which ranged from 20–700 μg L⁻¹. pH_e showed a slight increase in the mean at all depth increments from 1999 to 2004 followed by a leveling off from 2004 to 2009.

Table 2 Correlation coefficients between electromagnetic induction EC_a (both EM_h and EM_v) in $dS\ m^{-1}$ and soil properties measured over 0 to 1.2 m for 1999, 2002, 2004, 2009, and 2011. $N = 40^f$

	1999 ^c			2002 ^d			2004 ^d			2009			2011		
	EM_h , EC_a	EM_v , EC_a	EM_h , EC_a	EM_h , EC_a	EM_v , EC_a	EM_h , EC_a	EM_h , EC_a	EM_v , EC_a	EM_h , EC_a	EM_h , EC_a	EM_v , EC_a	EM_h , EC_a	EM_h , EC_a	EM_v , EC_a	EM_h , EC_a
θ_v (m^3/m^3)	0.62 ^{b,e}	0.64 ^{b,e}	0.59 ^b	0.63 ^b	0.55 ^b	0.54 ^b	0.70 ^b	0.87 ^b	0.75 ^b	0.77 ^b	0.89 ^b	0.77 ^b	0.87 ^b	0.89 ^b	0.89 ^b
EC_e ($dS\ m^{-1}$)	0.74 ^b	0.78 ^b	0.89 ^b	0.84 ^b	0.90 ^b	0.90 ^b	0.85 ^b	0.87 ^b	0.85 ^b	0.87 ^b	0.89 ^b	0.85 ^b	0.87 ^b	0.89 ^b	0.89 ^b
pH _e	0.01	-0.01	0.40 ^b	0.43 ^b	0.55 ^b	0.58 ^b	0.59 ^b	0.59 ^b	0.34 ^a	0.34 ^a	0.40 ^a	0.34 ^a	0.34 ^a	0.40 ^a	0.40 ^a
SP (%)	0.25	0.22	0.22	0.30	0.22	0.19	0.27	0.27	0.25	0.27	0.30	0.25	0.30	0.32 ^a	0.32 ^a
Cl ⁻ (mmol _e L ⁻¹)	0.34 ^a	0.43 ^b	0.77 ^b	0.81 ^b	0.80 ^b	0.82 ^b	0.83 ^b	0.83 ^b	0.79 ^b	0.83 ^b	0.83 ^b	0.83 ^b	0.83 ^b	0.79 ^b	0.79 ^b
SAR	0.64 ^b	0.70 ^b	0.84 ^b	0.82 ^b	0.87 ^b	0.88 ^b	0.84 ^b	0.84 ^b	0.81 ^b	0.84 ^b	0.85 ^b	0.81 ^b	0.85 ^b	0.87 ^b	0.87 ^b
B (mg L ⁻¹)	0.48 ^b	0.42 ^b	0.43 ^b	0.40 ^b	0.68 ^b	0.59 ^b	0.70 ^b	0.70 ^b	0.66 ^b	0.73 ^b	0.73 ^b	0.66 ^b	0.73 ^b	0.73 ^b	0.73 ^b
Se ($\mu g\ L^{-1}$)	0.56 ^b	0.51 ^b	NM	NM	-0.14	-0.15	-0.33 ^a	-0.33 ^a	-0.33 ^a	-0.33 ^a	-0.34 ^a	-0.33 ^a	-0.34 ^a	-0.34 ^a	-0.34 ^a
Mo ($\mu g\ L^{-1}$)	0.58 ^b	0.45 ^b	0.45 ^b	0.40 ^b	-0.13	-0.27	0.49 ^b	0.49 ^b	0.47 ^b	0.49 ^b	0.49 ^b	0.47 ^b	0.49 ^b	0.49 ^b	0.49 ^b

^a Significant (test for $|r| = 0$) at $p \leq 0.05$ level. ^b Significant (test for $|r| = 0$) at $p \leq 0.01$ level. ^c Taken from Corwin *et al.* (2003). ^d Taken from Corwin *et al.* (2008). ^e $N = 31$. ^f EC_a = apparent soil electrical conductivity, EM_h = electromagnetic induction measured in the horizontal dipole mode, EM_v = electromagnetic induction measured in the vertical dipole mode, θ_v = volumetric water content, EC_e = electrical conductivity of saturation extract, sp. = saturation percentage, SAR = sodium adsorption ratio.

Table 3 Mean and range statistics for 0.0–0.3 m sample depth for 1999, 2004, 2009, and 2011. $N = 48$ for each sample year^c

Soil property	1999 ^a			2004 ^b			2009			2011		
	Mean	Min.	Max.	Mean	Min.	Max.	Mean	Min.	Max.	Mean	Min.	Max.
θ_v (m^3/m^3)	0.30	0.21	0.48	0.32	0.22	0.39	0.32	0.22	0.39	0.32	0.22	0.39
EC_e ($dS\ m^{-1}$)	13.0	5.6	35.7	13.0	5.6	35.7	13.0	5.6	35.7	13.0	5.6	35.7
pH _e	7.6	6.9	8.3	7.6	6.9	8.3	7.6	6.9	8.3	7.6	6.9	8.3
SP (%)	58.8	45.9	79.1	58.8	45.9	79.1	58.8	45.9	79.1	58.8	45.9	79.1
Cl ⁻ (mmol _e L ⁻¹)	21.8	3.29	67.3	21.8	3.29	67.3	21.8	3.29	67.3	21.8	3.29	67.3
SAR	28.2	8.3	70.2	28.2	8.3	70.2	28.2	8.3	70.2	28.2	8.3	70.2
B (mg L ⁻¹)	17.0	1.1	42.5	17.0	1.1	42.5	17.0	1.1	42.5	17.0	1.1	42.5
Se ($\mu g\ L^{-1}$)	8.8	0.0	77.0	8.8	0.0	77.0	8.8	0.0	77.0	8.8	0.0	77.0
Mo ($\mu g\ L^{-1}$)	862.3	442.0	3043.0	862.3	442.0	3043.0	862.3	442.0	3043.0	862.3	442.0	3043.0

^a Taken from Corwin *et al.* (2003). ^b Taken from Corwin *et al.* (2008). ^c θ_v = volumetric water content, EC_e = electrical conductivity of saturation extract, SP = saturation percentage, SAR = sodium adsorption ratio.

Table 4 Mean and range statistics for 0.3–0.6 m sample depth for 1999, 2004, 2009, and 2011. $N = 48$ for each sample year^c

Soil property	1999 ^a					2004 ^b					2009					2011								
	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.
θ_v (m ³ /m ³)	0.40	0.29	0.52	0.05	11.8	0.36	0.34	0.29	0.41	0.03	7.3	0.44	0.35	0.24	0.47	0.06	17.1	0.25	0.29	0.17	0.47	0.066	22.8	1.18
EC _e (dS m ⁻¹)	20.2	13.5	34.5	5.3	26.0	1.05	16.5	4.38	35.5	7.73	46.7	0.55	15.0	3.64	32.0	7.35	49.0	0.60	22.0	5.66	46.6	9.85	44.8	0.39
pH _e	7.6	7.0	8.0	0.2	2.6	-0.89	7.6	6.7	8.3	0.41	5.4	-0.44	7.7	7.3	8.0	0.16	2.1	-0.38	7.9	7.4	8.4	0.18	2.3	0.086
SP (%)	63.0	45.5	84.1	10.2	16.2	0.01	65.1	44.3	98.9	12.0	18.4	0.50	64.5	55.4	83.0	9.01	14.0	0.08	57.4	40.5	76.5	8.66	15.1	0.19
Cl ⁻ (mmol _c L ⁻¹)	35.3	12.9	69.0	14.3	40.6	0.50	31.6	4.92	106.0	20.1	63.7	1.20	26.2	4.04	95.4	18.2	69.4	1.71	59.6	5.03	166.3	34.6	58.1	0.64
SAR	51.4	30.3	89.5	12.9	25.1	0.87	41.3	5.2	100.2	21.3	51.4	0.41	37.6	4.32	90.2	20.1	53.5	0.53	47.3	8.02	105.4	22.6	47.8	0.28
B (mg L ⁻¹)	19.0	13.6	38.1	5.6	29.7	1.90	14.2	1.39	33.7	7.50	53.0	0.78	12.1	1.14	30.4	6.55	54.1	1.03	14.7	1.94	48.7	10.2	69.4	1.49
Se (μg L ⁻¹)	14.0	0.0	45.0	8.5	60.9	0.79	77.0	0.0	529.0	140.0	181.0	2.06	62.0	0.0	410.0	110.0	177.4	2.13	1438.7	0.0	25 574	4673.8	324.8	4.22
Mo (μg L ⁻¹)	750.5	180.0	2488.0	430.2	57.3	2.63	360.0	0.0	1589.0	343.0	95.4	1.57	276.8	0.0	1367.0	278.9	100.8	2.00	596.7	62.9	2021.9	379.6	63.6	1.93

^a Taken from Corwin *et al.* (2003). ^b Taken from Corwin *et al.* (2008). ^c θ_v = volumetric water content, EC_e = electrical conductivity of saturation extract, SP = saturation percentage, SAR = sodium adsorption ratio.

Table 5 Mean and range statistics for 0.6–0.9 m sample depth for 1999, 2004, 2009, and 2011. $N = 48$ for each sample year^c

Soil property	1999 ^a					2004 ^b					2009					2011								
	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.
θ_v (m ³ /m ³)	0.40	0.23	0.51	0.05	12.4	-0.72	0.36	0.27	0.41	0.03	8.7	-0.84	0.36	0.17	0.47	0.06	16.7	-0.54	0.32	0.12	0.46	0.07	21.9	-0.47
EC _e (dS m ⁻¹)	22.5	9.7	43.2	6.5	28.7	1.27	21.5	4.65	48.3	10.0	46.6	0.49	19.5	3.86	43.5	9.66	49.5	0.62	26.5	6.57	44.0	8.72	32.9	-0.24
pH _e	7.6	7.3	8.1	0.2	2.3	0.32	7.8	7.1	8.4	0.33	4.2	-0.26	7.8	7.4	8.3	0.20	2.6	0.16	8.0	7.5	8.4	0.25	3.1	-0.20
SP (%)	59.1	40.0	89.5	11.1	18.8	1.16	62.2	40.9	95.8	12.8	20.6	0.69	59.9	40.4	93.4	11.7	19.5	1.11	58.3	40.4	95.0	11.1	19.0	1.19
Cl ⁻ (mmol _c L ⁻¹)	47.1	5.48	100.5	21.8	46.3	0.68	46.6	6.19	154.0	32.3	69.4	1.11	35.3	5.19	101.6	21.9	62.0	0.89	74.1	10.8	144.2	30.2	40.8	0.090
SAR	59.0	24.0	107.6	16.6	28.1	0.76	56.9	6.3	121.0	26.5	46.5	0.09	51.9	5.25	108.9	24.9	48.0	0.23	60.2	10.9	100.1	21.2	35.2	-0.35
B (mg L ⁻¹)	17.5	9.4	31.3	4.8	27.2	0.87	16.7	1.64	37.2	7.28	43.6	0.39	14.3	1.35	33.5	6.36	44.4	0.68	17.9	2.69	42.5	7.94	44.4	0.71
Se (μg L ⁻¹)	12.9	0.0	34.0	9.2	71.5	0.67	70.0	0.0	704.0	141.0	203.3	2.67	64.0	0.0	550.0	120.0	187.5	2.31	483.9	0.0	6191.3	1361.1	281.3	3.28
Mo (μg L ⁻¹)	780.5	183.0	1756.0	338.9	43.4	0.90	294.0	0.0	169.0	350.0	119.3	1.74	215.7	0.0	1453.0	299.3	138.8	2.30	481.7	0.0	1663.5	330.9	68.7	1.59

^a Taken from Corwin *et al.* (2003). ^b Taken from Corwin *et al.* (2008). ^c θ_v = volumetric water content, EC_e = electrical conductivity of saturation extract, SP = saturation percentage, SAR = sodium adsorption ratio.

Table 6 Mean and range statistics for 0.9–1.2 m sample depth for 1999, 2004, 2009, and 2011. $N = 48$ for each sample year^c

Soil property	1999 ^a						2004 ^b						2009						2011					
	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.	Mean	Min.	Max.	SD	CV	Skew.
θ_v (m^3/m^3)	0.43	0.31	0.54	0.05	11.4	-0.37	0.36	0.27	0.44	0.04	10.3	-0.55	0.39	0.22	0.58	0.07	17.9	-0.19	0.35	0.15	0.62	0.091	26.0	0.062
EC_e (dS m^{-1})	25.2	8.0	49.7	7.9	31.5	0.81	22.5	4.33	47.4	9.80	43.5	0.24	20.2	3.59	42.7	9.32	46.1	0.39	27.4	7.41	52.1	8.48	30.9	-0.008
pH_e	7.6	7.1	8.0	0.2	2.7	-0.37	7.9	7.0	8.4	0.31	3.9	-0.79	7.8	7.3	8.2	0.20	2.6	-0.40	8.1	7.6	8.5	0.22	2.7	0.094
SP (%)	58.7	38.5	93.3	12.9	22.0	0.68	64.2	38.4	134.1	17.5	27.2	1.46	61.2	38.2	102.0	13.4	21.9	0.72	60.6	38.0	108.7	13.8	22.7	1.27
Cl^- ($\text{mmol}_e \text{ L}^{-1}$)	58.7	3.18	152.7	30.2	51.4	0.80	51.7	6.40	129.7	32.0	62.0	0.40	41.1	5.31	116.7	26.9	65.5	0.75	75.7	15.3	162.8	31.2	41.2	0.17
SAR	64.9	16.8	120.2	19.5	30.0	0.35	60.9	5.4	127.8	26.5	43.6	0.05	55.0	4.47	115.0	24.6	44.7	0.23	64.1	12.7	98.9	20.7	32.3	-0.49
B (mg L^{-1})	17.9	6.5	31.8	6.3	35.0	0.45	16.5	1.33	34.5	7.32	44.5	0.15	14.4	1.09	31.0	6.54	45.4	0.39	18.3	2.96	37.8	7.38	40.3	0.15
Se ($\mu\text{g L}^{-1}$)	14.1	0.0	62.0	13.9	98.5	1.80	81.0	0.0	677.0	161.0	199.8	2.62	62.0	0.0	550.0	140.0	225.8	2.70	374.9	0.0	5361.6	1059.7	282.7	3.42
Mo ($\mu\text{g L}^{-1}$)	946.9	330.0	2856.0	450.7	47.6	1.78	278.0	0.0	1159.0	272.0	98.0	0.90	212.2	0.0	997.0	224.9	106.0	1.26	412.4	0.0	1151.0	271.7	65.9	1.22

^a Taken from Corwin *et al.* (2003). ^b Taken from Corwin *et al.* (2008). ^c θ_v = volumetric water content, EC_e = electrical conductivity of saturation extract, SP = saturation percentage, SAR = sodium adsorption ratio.

Vertical profiles for EC_a and SAR from 1999 to 2009 showed an increase with depth, which is typical of leaching in the presence of a plant (Fig. 3a and b). Boron also increased with depth (see Fig. 3c), but leveled off at the bottom depths, 0.6–0.9 and 0.9–1.2 m, which is likely the consequence of the strong adsorption of B and insufficient drainage water application to leach B beyond 0.9 m. In slight contrast to the vertical profiles of EC_e , SAR and B, Mo was high near the surface (0–0.3 m), then dropped to its lowest level and increased with depth (see Fig. 3d), which points to a recycling of the Mo from the root zone to the soil surface by plants. Salinity, as measured by EC_e , decreased at all depths from 1999 to 2009 (Fig. 3a), resulting in an 21% decrease in salinity from 1999 to 2009 in the top 1.2 m soil profile on a mass basis. A comparable 19% decrease occurred for SAR from 1999 to 2009 with the sharpest decrease from 1999 to 2002 (Fig. 3b). Boron decreased 32% from 1999 to 2009 (Fig. 3c) and Mo decreased by 67% from 1999 to 2009 for the 1.2 m soil profile (Fig. 3d).

Adsorption of B and Mo is strongly dependent on pH (Goldberg *et al.*, 2002, 2005).^{35,36} The maximum adsorption for B occurs around pH 9 (Goldberg *et al.*, 2005).³⁶ The maximum adsorption for Mo occurs in the pH range 2–4 with rapid decrease in adsorption with increasing pH (Goldberg *et al.*, 2002).³⁵ For the pHs of the soil in this study, B was more strongly adsorbed and was less mobile than Mo (Fig. 3c and d). However, even though B was strongly adsorbed, the adsorption capacity was not great; consequently, B was steadily leached from the upper depths of the soil profile (top 0.6 m) from 1999 to 2009 (Fig. 3c). Molybdenum was very mobile. Each successive sampling from 1999 to 2009 showed a decrease with greater leaching of Mo occurring below 0.3 m (Fig. 3d). The slower removal of Mo from the top 0.3 m suggests bypass or greater retention due to adsorption. The change in field means from 1999 to 2004 for Cl^- at the top two depth increments (see Tables 3 and 4) suggests that bypass may be occurring as indicated by the greater removal of Cl^- from the 0.3–0.6 m depth increment than from the 0–0.3 depth increment, but this condition did not continue from 2004 to 2009. Furthermore, Na and salinity did not show any sign of bypass; consequently, bypass in the top 0.3 m was ruled out. Rather, the recycling of Mo by plant roots is a likely explanation. Since about 2002, an invasion of sweet clover (*Melilotus*), which contained high concentrations of Mo in tissue samples, may have recycled Mo.

Table 7 shows the significance of changes by depth for shift in the field means over time for EC_e , SAR, B, Mo, and pH_e . In every case except for pH_e at the 0–0.3 m depth increment there was a significant difference between the field means of 1999 and 2009, confirming the impact of drainage water reuse on the saline-sodic field. The general temporal trend from 1999 to 2009 for the field means of pH_e was a gradual increase at each depth increment, but pH_e actually decreased or remained the same from 2004 to 2009 (see Tables 3–6). Even though field means are of value to understand general trends that are occurring for the entire field, they provide no spatial information to determine where areas of greatest concern may or may not exist within the field. The change in pH_e is a good example and the following section discussing spatio-temporal trends will shed more light on the observed trends in pH_e .

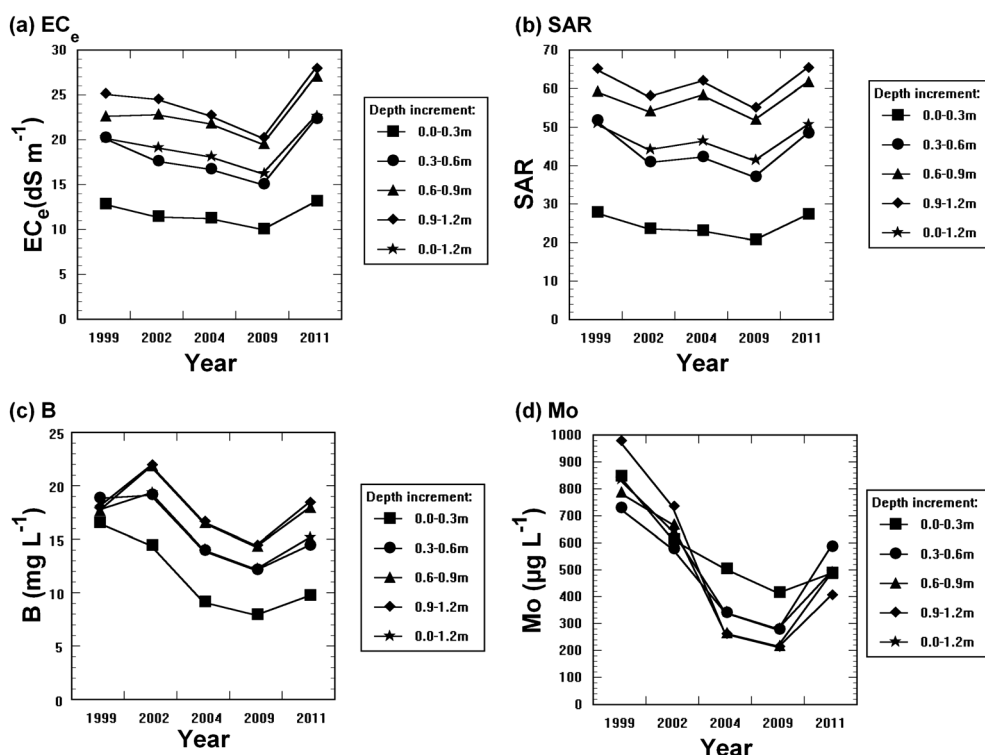


Fig. 3 Graphs of (a) EC_e (salinity), (b) SAR, (c) B, and (d) Mo showing the change of the field means for the depth increments 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m and the composite depth 0–1.2 m for the years 1999, 2002, 2004, 2009, and 2011. EC_e , electrical conductivity of the saturation extract ($dS\ m^{-1}$); SAR, sodium adsorption ratio (dimensionless). Year 1999 is taken from Corwin *et al.* (2003)⁶ and years 2002 and 2004 are taken from Corwin *et al.* (2008).⁷

The coefficient of variation (CV) is a normalized measure of dispersion of a probability distribution indicating the field-scale variation. On a relative basis for the soil properties of interest, CVs for θ_v , pH_e , and SP are low (*i.e.*, $CV < 30$) and CVs for Se and Mo are high (*i.e.*, $CV > 70$), while CVs for EC_e , Cl^- , SAR and B are moderate, regardless of the year the soil sample was taken.

Spatio-temporal trends of EC_e , SAR, B, Mo, Se, and pH_e during drainage water reuse (1999–2009)

The trends that occurred over time are readily understood for the field as a whole from the graphs in Fig. 3 and Tables 3–6, but the comparison of field means without knowledge of spatial distribution can be misleading. In the subsequent discussion spatial changes were gleaned from maps shown in Fig. 4–9, which provide vertical and horizontal changes in distribution.

To determine if there was change in the spatial variation across the field during the 12 years of the study, which would indicate that additional monitoring sites may be needed beyond the 40 sites, the *F*-test for dynamic spatial variation was used as described in Corwin *et al.* (2006).²⁹ Table 7 shows the significance levels (*p*-values) by depth increment corresponding to the *F*-test for no dynamic variation ($\Phi_{is}^2 = 0$) for EC_e , SAR, B, Mo, and pH_e . Data taken from Corwin *et al.* (2008)⁷ for the extent of dynamic spatial variation for EC_e , SAR, B, and Mo from 1999 to 2004 (see Table 7) indicate that future monitoring efforts should consider the addition of new monitoring sites to characterize

temporal changes. This is not the case for 1999 to 2009 and the dynamic spatial variation from 2009 to 2011 shows only that additional monitoring sites are needed to characterize Mo spatially. Even though monitoring sites were initially added for the 2009 and 2011 soil sampling based on the recommendation of Corwin *et al.* (2008),⁷ they were not deemed necessary and were not included in this paper so that comparisons of an equal number of samples were made. Ostensibly, the general patterns of spatial distribution for EC_e , SAR, and B are similar over time due to the general water flow patterns and chemical reactivity that are influenced by textural distribution, which is a static property. The dynamic spatial variation of Mo over time indicates that there are additional interactions that are coming into play such as the recycling of Mo by plant roots due to the invasion of sweet clover (*Melilotus*) in 2002. Sweet clover showed high concentrations of Mo in tissue samples that would definitely have an influence on the vertical distribution of Mo but it is difficult to conceptualize how this would also have a substantial influence on the areal distribution. The dissimilar spatial behavior of Mo over time compared to EC_e , SAR, and B cannot be explained with the physical and chemical data collected in this study.

Spatio-temporal changes in distribution for EC_e , SAR, B, and Mo from 1999 to 2011 are shown in Fig. 4a–c to 7–c, respectively. Each figure contains 3 sets of 4 maps corresponding to the sampling times of 1999, 2009, and 2011 and four depth increments (*i.e.*, 0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m). Each map provides an areal distribution of the soil property at a specified

Table 7 *F*-Test significance levels associated with mixed linear modeling results^b

Chemical property	Sample depth increment (m)	Pr > F for shift in mean level (2009 vs. 1999)	Pr > F for shift in mean level (2011 vs. 2009)	Pr > F for dynamic spatial variation (2004 vs. 1999) ^a	Pr > F for dynamic spatial variation (2009 vs. 1999)	Pr > F for dynamic spatial variation (2011 vs. 2009)
EC _e	0–0.3	0.0032	0.0001	0.0001	0.1700	0.0296
	0.3–0.6	0.0001	0.0001	0.0037	0.0493	0.1291
	0.6–0.9	0.0012	0.0001	0.0024	0.0033	0.0696
	0.9–1.2	0.0001	0.0001	0.0070	0.0082	0.2344
SAR	0–0.3	0.0037	0.0012	0.0001	0.1240	0.0263
	0.3–0.6	0.0001	0.0003	0.0009	0.0561	0.1076
	0.6–0.9	0.0025	0.0009	0.0031	0.0023	0.0730
	0.9–1.2	0.0019	0.0005	0.0006	0.0003	0.2422
B	0–0.3	0.0001	0.0024	0.0320	0.3874	0.3675
	0.3–0.6	0.0001	0.0032	0.0005	0.0501	0.0322
	0.6–0.9	0.0433	0.0016	0.0006	0.0586	0.0402
	0.9–1.2	0.0509	0.0009	0.0015	0.0012	0.0947
Mo	0–0.3	0.0001	0.0497	0.3472	0.1547	0.1879
	0.3–0.6	0.0001	0.0001	0.0006	0.0001	0.0291
	0.6–0.9	0.0001	0.0001	0.0031	0.0006	0.0456
	0.9–1.2	0.0001	0.0192	0.0247	0.0008	0.0368
pH _e	0–0.3	0.0822	0.0751	0.5576	0.7784	0.9020
	0.3–0.6	0.0488	0.0222	0.0040	0.6886	0.1783
	0.6–0.9	0.0027	0.0011	0.0168	0.1993	0.3877
	0.9–1.2	0.0016	0.0001	0.0030	0.0499	0.7593

^a Taken from Corwin *et al.* (2008).⁷ ^b EC_e = electrical conductivity of saturation extract, SAR = sodium adsorption ratio.

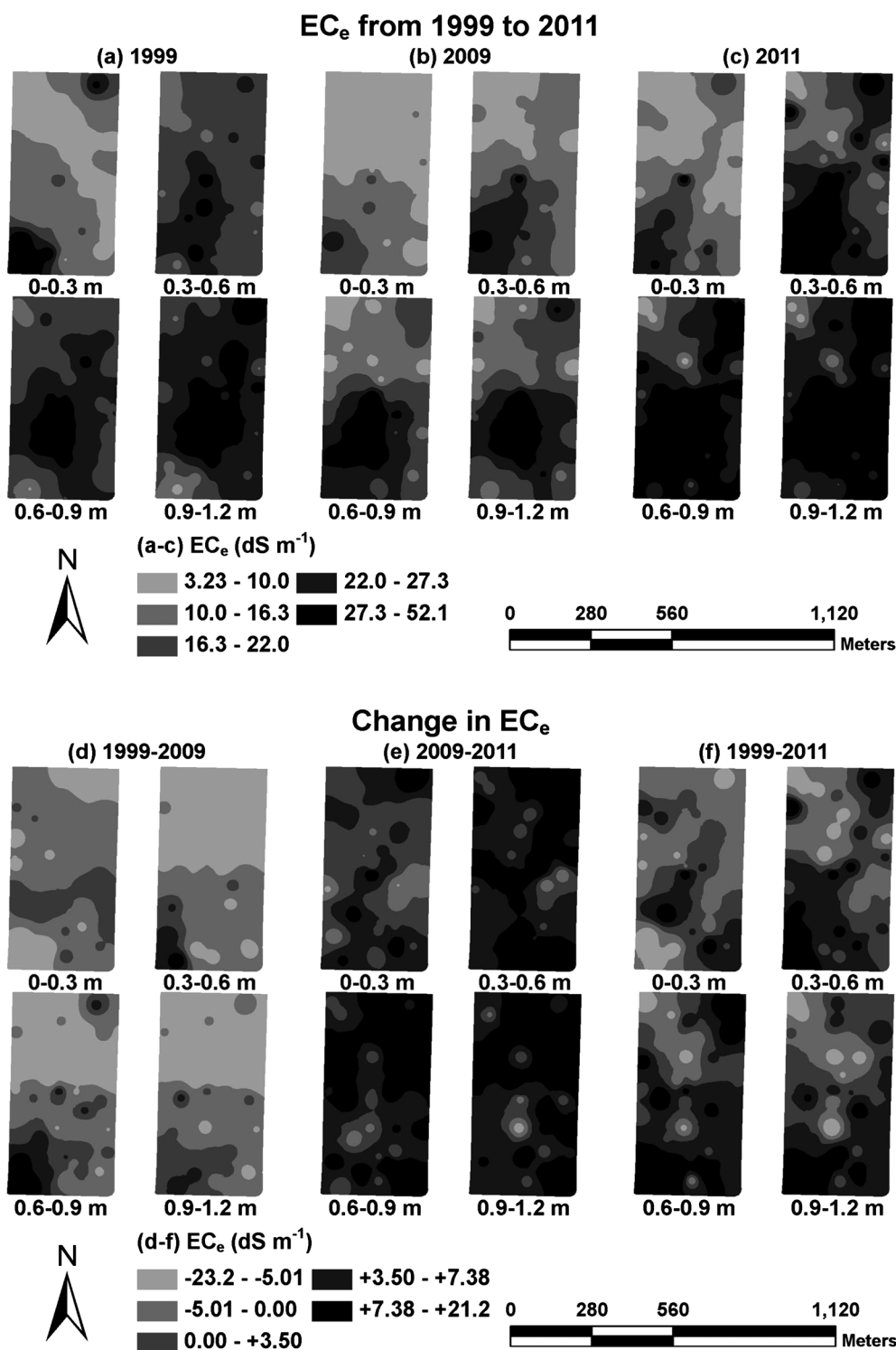


Fig. 4 Maps showing the change in spatial patterns of salinity (electrical conductivity of the saturation extract, EC_e) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2009, and (c) 2011. Maps showing spatial patterns of the net change of EC_e from (d) 1999–2009, (e) 2009–2011, and (f) 1999 to 2011 by depth increment. Data for year 1999 taken from Corwin *et al.* (2003).⁶

depth increment and time. Each set of 4 maps provides a snapshot in time of the property's 3-dimensional distribution. The last 3 sets of maps in each figure (*i.e.*, Fig. 4d–f, 5d–f, 6d–f, and 7d–f) display the net change in EC_e, SAR, B, and Mo from 1999 to 2009, 2009 to 2011, and over the full 12 years of the study (*i.e.*,

1999 to 2011), with light areas indicating a net loss and dark areas indicating a net gain. Fig. 4–7 provide a visual means of evaluating the spatial baselines, spatio-temporal trends, and spatial net changes for EC_e, SAR, B, and Mo during irrigation with drainage water from 2000 to 2009 and during the time

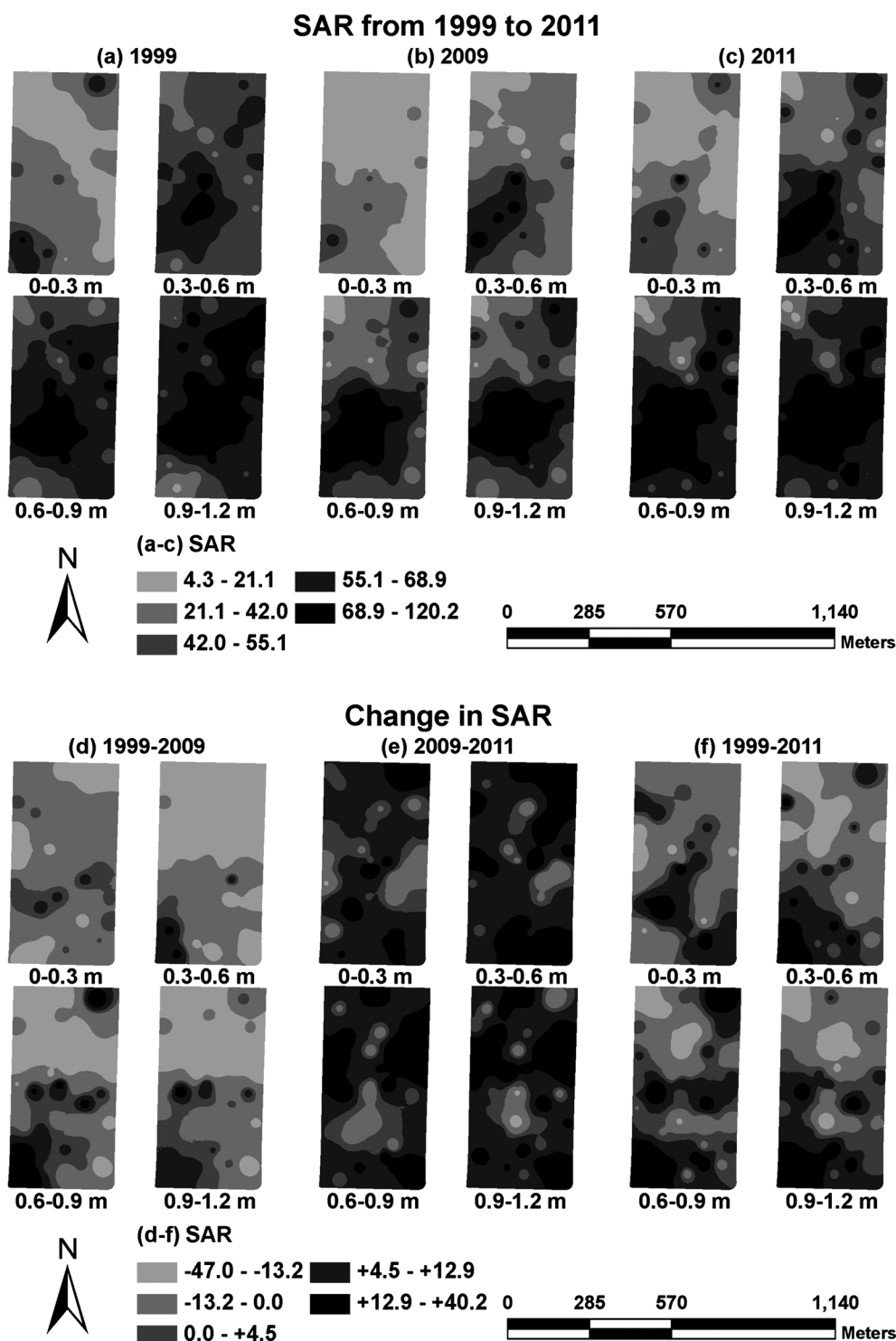


Fig. 5 Maps showing the change in spatial patterns of SAR (sodium adsorption ratio) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2009, and (c) 2011. Maps showing spatial patterns of the net change of SAR from (d) 1999–2009, (e) 2009–2011, and (f) 1999 to 2011 by depth increment. Data for year 1999 taken from Corwin *et al.* (2003).⁶

period when no irrigation water was applied and only precipitation occurred, *i.e.*, 2009 to 2011.

General similarities are evident in the 1999 baseline spatial patterns of EC_e, SAR, B and Mo (see Fig. 4a, 5a, 6a, and 7a, respectively), suggesting that these patterns are the consequence

of a common process, most likely water flow through the root zone (*i.e.*, 0–1.2 m). As indicated by Corwin *et al.* (2008),⁷ a general spatial pattern of higher levels of EC_e, SAR, B, and Mo in the south than the north existed from 1999 to 2004, suggesting that high levels of Na in the southern half of the field, particularly

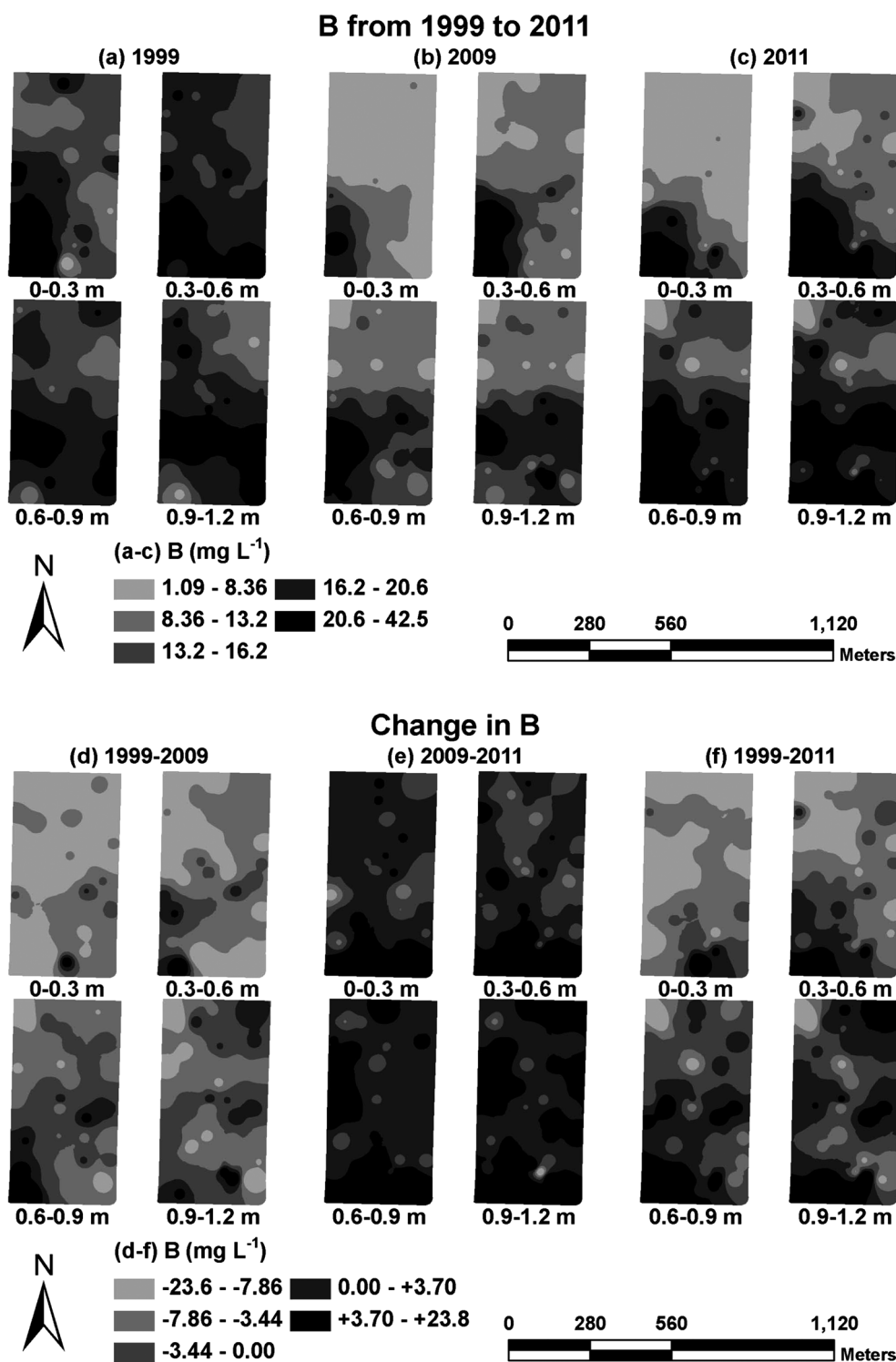


Fig. 6 Maps showing the change in spatial patterns of B by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2009, and (c) 2011. Maps showing spatial patterns of the net change of B from (d) 1999–2009, (e) 2009–2011, and (f) 1999 to 2011 by depth increment. Data for year 1999 taken from Corwin *et al.* (2003).⁶

in the southwest, dispersed the soil, which restricted infiltration and subsequent leaching, resulting in the trend of high to low levels of EC_e, SAR, B, and Mo from south to north. This general north–south pattern with particularly high levels in the southwest held for EC_e, SAR, and B from 1999–2009 (see Fig. 4a–c,

5a–c, 6a–c, and 7a–c), with changes occurring primarily in magnitude as a result of the leaching process. Corwin *et al.* (2008)⁷ noticed a reappearance of Mo in the north in 2004, which continued to accumulate up to 2009, but not to any appreciable extent since the overall change in the north from 1999 to 2009

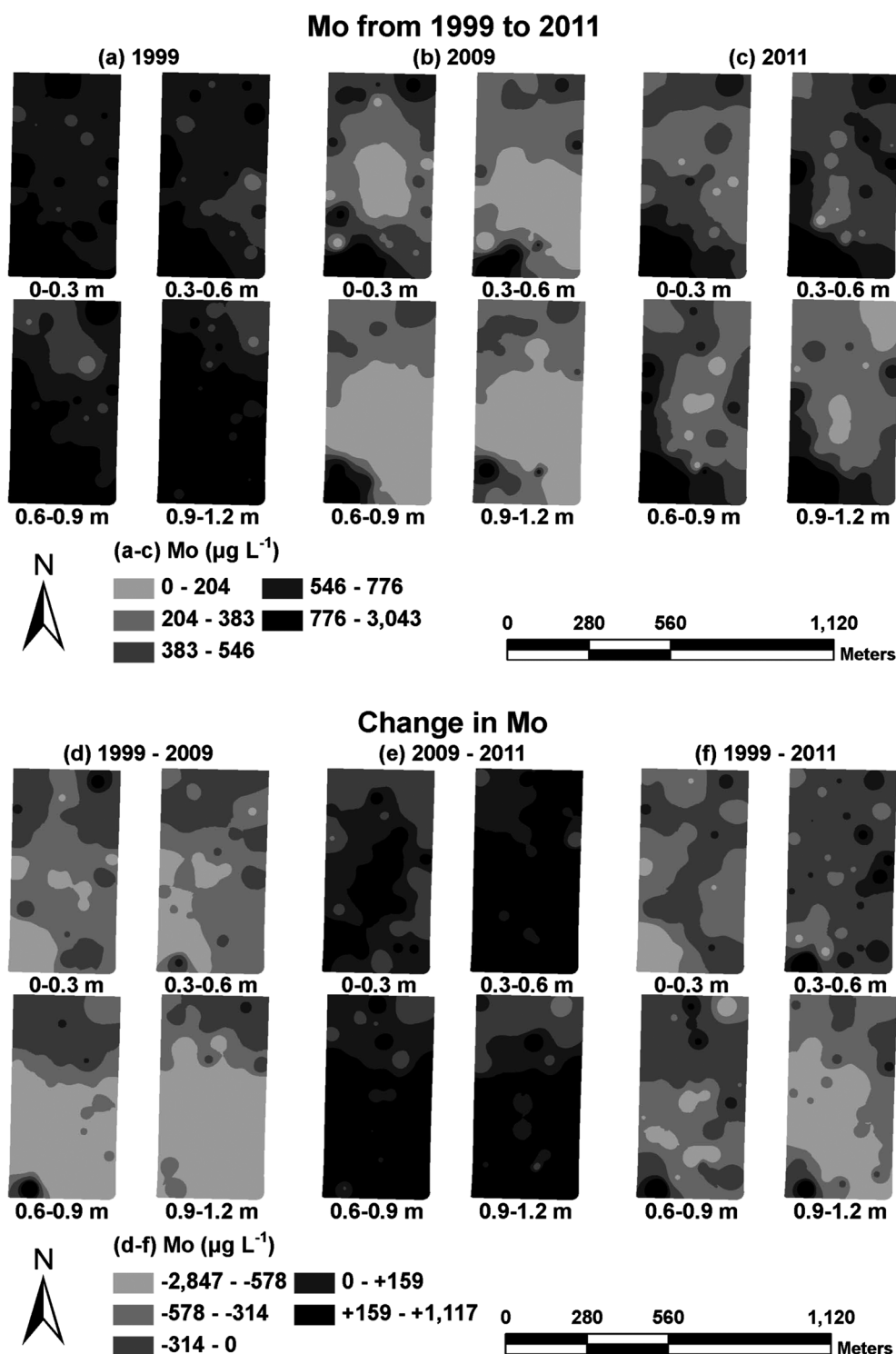


Fig. 7 Maps showing the change in spatial patterns of Mo by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2009, and (c) 2011. Maps showing spatial patterns of the net change of Mo from (d) 1999–2009, (e) 2009–2011, and (f) 1999 to 2011 by depth increment. Data for year 1999 taken from Corwin *et al.* (2003).⁶

was a net loss of Mo (see Fig. 7d) due to the substantial leaching of Mo from 1999 to 2002.

Spatio-temporal changes of EC_e and SAR from 1999 to 2009 (see Fig. 4d and 5d) are very similar, which is the consequence of the dominance of Na^+ salts to the total salinity. The changes that

occur in EC_e and SAR from 1999 to 2009 show a general decrease in EC_e through the entire profile 0–1.2 m. This is substantiated by the *F*-test significance levels of EC_e and SAR in Table 7, which show that the shift in mean level is significant for each depth increment in the top 1.2 m. Even though the general

spatial patterns appear stable from 1999 to 2009 in Fig. 4a and b and 5a and b, there are definite spatial changes occurring. Specifically, Fig. 4d and 5d indicate that leaching of salts and Na is occurring to a greater extent in the northern half of the field, with removal at all depths, while in the southern half and especially in the southwest corner there is an accumulation of salts and Na in the lower depths (*i.e.*, 0.6–1.2 m). This pattern is most readily explained by the dispersion effect due to high SAR in the south, which creates infiltration and permeability problems. Even though the decrease in EC_e and SAR is not as dramatic in the south as the north, it is readily apparent that salts and Na are being leached throughout the field to a significant extent and from 1999 to 2009 the saline-sodic soil is being reclaimed.

From 1999 to 2009 the spatial net changes for B and Mo show definite dissimilarities (Fig. 6d and 7d). Spatio-temporal trends from 1999 to 2009 show more leaching of B occurs in the north than in the south with B accumulating at the bottom two depths (0.6–0.9 and 0.9–1.2 m) in the south. Molybdenum is observably more mobile than B with a greater proportion of Mo leached from all depth increments and greater leaching of Mo occurring in the south than in the north, particularly at the bottom depths (0.6–0.9 and 0.9–1.2 m). The greater removal of Mo from the bottom depths once again suggests a recycling of Mo by plant roots from the bottom of the root zone to the soil surface. Whereas, the movement of B in the south is retarded due to the high Na levels that disperse the soil and slow water flow. The *F*-test significance levels for shift in field means from 1999 to 2009 show that there was a significant ($p \leq 0.05$) decrease in B concentration only in the top 0.9 m, whereas a significant decrease in Mo concentration occurred for the entire 1.2 m soil profile (Table 7).

Aside from concern over the reappearance of Mo, Corwin *et al.* (2008)⁷ also warned of the accumulation of Se. Concentrations of Se in 2004 were found as high as $704 \mu\text{g L}^{-1}$ and average root zone levels ranged from $57\text{--}81 \mu\text{g L}^{-1}$. However, by 2009 the field means of Se had decreased at all depths from 2004 levels (see Tables 3–6), which coincides with the decrease in Se concentration in the drainage water applied to the field. After 29 Aug. 2004 the Se concentration in the applied drainage water was not above $1 \mu\text{g L}^{-1}$ (Table 1). Even though Tables 3–6 indicate that Se was leached after 2004, Fig. 8a shows that there was substantial increase in Se at the north and south ends with an east–west band in the middle of the field where Se decreased, suggesting the need for continued spatial monitoring.

Corwin *et al.* (2008)⁷ expressed concern over the rise in pH_e during the first few years of applying drainage water. However, from 1999 to 2009 only a slight increase in pH_e occurred with very little change in the top 0.6 m and the greatest change occurring in the 0.9–1.2 m depth increment. The increase in pH_e in the top 0.6 m ranged from 0.05 to 0.1 pH units and the increase in pH_e below 0.6 m ranged from 0.05 to 0.25 pH units. The north–south differences that developed from 1999–2004 are no longer apparent in 2009 (Fig. 9b). Because the changes in pH_e are more subtle it is best to look at a succession of snapshots from 1999 to 2009 rather than look at the overall spatial change from 1999 to 2009. Fig. 10a–c provide a series of snapshots for 1999–2002, 2002–2004, and 2004–2009, respectively. It can be readily seen that the greatest increase in pH_e for all depths occurred from 1999 to 2002, particularly for the depths below 0.6 m (Fig. 10a). The increase in pH_e relates largely to the leaching of the saline–

sodic soil. Unlike saline soils, leaching of saline-sodic soils will increase the pH because once neutral salts are removed the exchangeable Na hydrolyzes and increases the OH^- concentration. In subsequent snapshots, *i.e.*, 2002–2004 and 2004–2009, there is a substantial portion of the field where pH_e decreases no matter what the depth is (Fig. 10b and c).

Impacts of the termination of drainage water reuse (2009–2011)

From June 2009 to May 2011 substantial changes occurred as a consequence of the termination of the reuse of drainage water. The field received rainfall as the sole source of water during this time period, which amounted to a total of 476 mm, most of which occurred in 2010 (333 mm). During this time period EC_e , SAR, B, Mo, Se, and pH_e increased at all depths, with EC_e and SAR, returning to levels near those of 1999 while Se and pH_e increased to above 1999 levels (see Fig. 3 and Tables 3–6). Fig. 3 graphically shows the abrupt increase in EC_e , SAR, B, and Mo from 2009 to 2011 for each depth increment. The significant increase in Se is particularly noteworthy in Tables 3–6. The *F*-test significance levels for shift in the field means from 2009 to 2011 (Table 7) confirms that there was a significant ($p \leq 0.05$) increase in EC_e , SAR, B, Mo, and pH_e at all depths except at 0–0.3 m for pH_e . Selenium showed a similar significant increase at all depths from 2009 to 2011 with *F*-test significance levels of 0.0001, 0.0001, 0.0046, and 0.0009 for the depth increments 0–0.3, 0.3–0.6, 0.6–0.9, 0.9–1.2 m, respectively. In just under 2 years following the termination of drainage water application the soil quality had nearly returned to its original poor quality.

Spatially, EC_e and SAR increased throughout the field from 2009 to 2011 particularly below 0.3 m (see Fig. 4e and 5e, respectively), with a conspicuous pocket of slightly decreasing EC_e and SAR in the southern half of the field for the bottom depths (*i.e.*, 0.6–0.9 and 0.9–1.2 m) where the highest levels of Na existed and permeability was the lowest. Boron and Mo showed a greater increase at all depths in the southern end of the field (see Fig. 6e and 7e, respectively). In contrast, Se increased more in the northern half of the field from 2009 to 2011, with decreases occurring in the southern half (see Fig. 8b). The pH_e increased throughout the field with greater increases occurring in the southern half where Na levels were highest (see Fig. 10d).

By 2011 the vertical profile of Se was inverted based on the field means (*i.e.*, field mean Se decreases with depth: $2824 \mu\text{g L}^{-1}$ at 0–0.3 m, $1439 \mu\text{g L}^{-1}$ at 0.3–0.6 m, $484 \mu\text{g L}^{-1}$ at 0.6–0.9 m, and $375 \mu\text{g L}^{-1}$ at 0.9–1.2 m), which is characteristic of the upward movement of solute from a shallow water table with accumulation at the soil surface (see Tables 3–6). In contrast, the field means for EC_e , SAR, B, and pH_e are regular profiles (*i.e.*, increase with depth), while Mo has an irregular profile with fluctuating concentrations through the soil profile (see Tables 3–6). The only source for salinity, Na, B, Mo, and Se is from the perched groundwater, which was just below the tile drains at 1.5 m; consequently, it can only be assumed that the high clay content of the soil and shallow water table resulted in the upward movement of these chemical constituents into the root zone. Clearly, on the WSJV downward flow of water is needed to maintain the soil quality once the soil has been reclaimed. Without downward flow the perched water table serves as a source of salts and trace elements, which return to the root zone

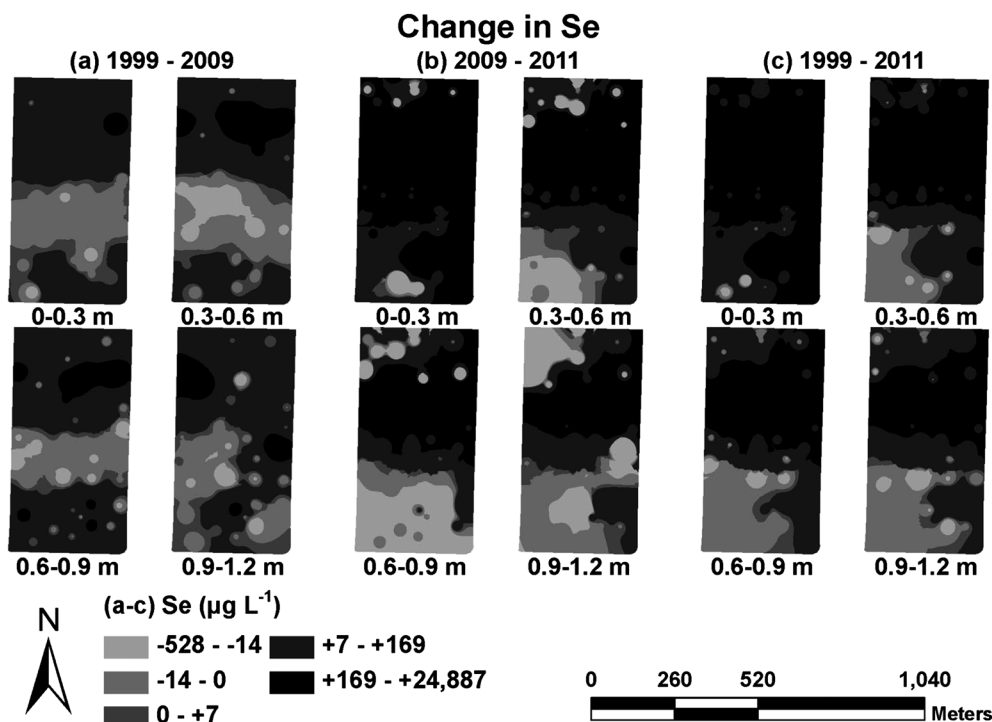


Fig. 8 Maps showing spatial patterns of the net change of Se from (a) 1999–2009, (b) 2009–2011, and (c) 1999 to 2011 by depth increment.

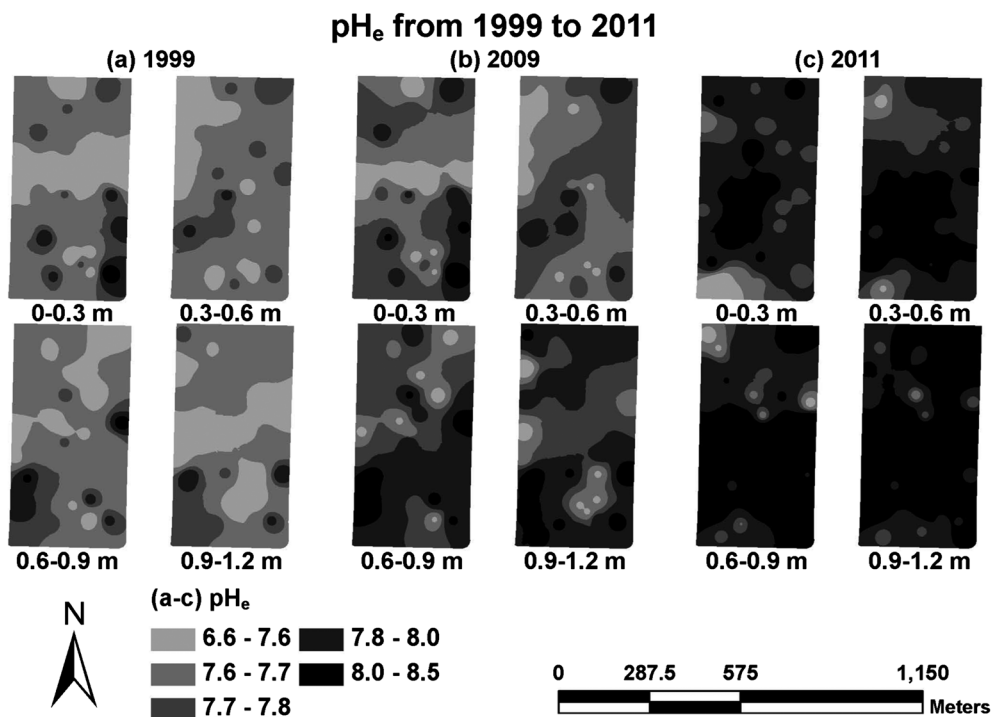


Fig. 9 Maps showing the change in spatial patterns of pH_e (pH of the saturation extract) by depth increment (0–0.3, 0.3–0.6, 0.6–0.9, and 0.9–1.2 m) for the sampling times of (a) 1999, (b) 2009, and (c) 2011. Data for year 1999 taken from Corwin *et al.* (2003).⁶

as a result of capillary rise of water. Even though only Se shows the characteristic inverted profile commonly associated with upward water flow from a shallow water table, there are no other plausible explanations.

4. Conclusions and implications

Unquestionably, and expectedly so, there are overlaps in the results presented in this paper and the work of Corwin *et al.*

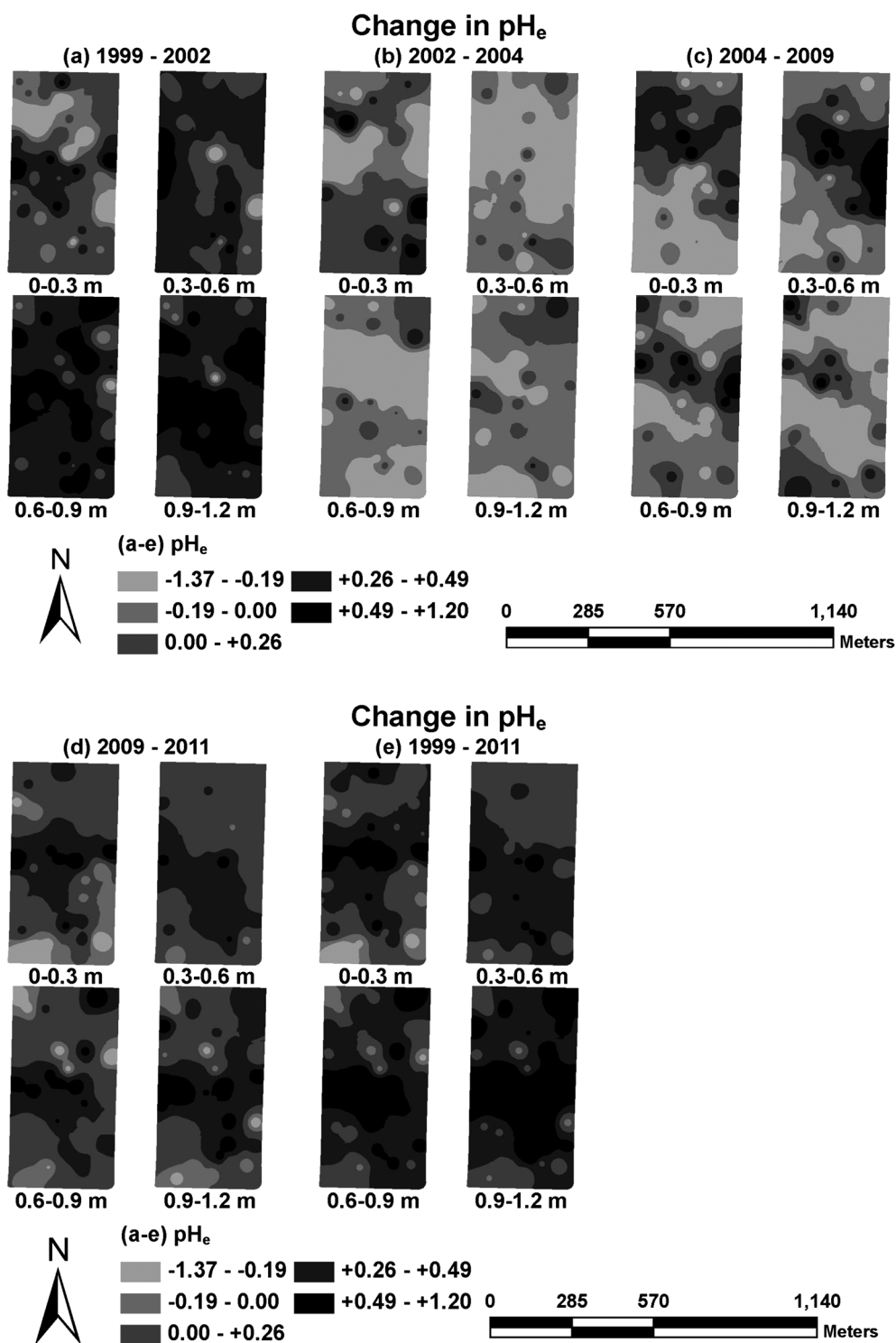


Fig. 10 Maps showing spatial patterns of the net change of pH_e (pH of the saturation extract) from (a) 1999–2002, (b) 2002–2004, (c) 2004 to 2009, (d) 2009 to 2011, and (e) 1999 to 2011 by depth increment. Data for year 1999 is taken from Corwin *et al.* (2003)⁶ and years 2002 and 2004 are taken from Corwin *et al.* (2008).⁷

(2008).⁷ However, there are distinct contributions that distinguish this long-term study from the previous short-term evaluation of the impacts of drainage water reuse on a saline-sodic soil. The most obvious is the confirmation of the long-term sustainability of drainage water reuse on a marginally productive

saline-sodic soil, which extends its viability from 5 years as demonstrated by Corwin *et al.* (2008)⁷ to 10 years. Demonstration of the long-term sustainability of this approach provides the credibility necessary for technological transfer to real-world application. This study also revokes the concern of Corwin *et al.*

(2008)⁷ that trace elements can accumulate during drainage water reuse in the WSJV posing a potential threat to the quality and yield of the crop grown. Not only was the saline–sodic soil reclaimed through the reuse of drainage water, but the soil was returned to productivity in 2 years, producing forage for livestock with a net profit to the producer. Even though the present long-term study has shown that trace element accumulation from drainage water reuse is manageable for California's WSJV soil, it is advisable to consider the chemical constituents in the soil that could potentially build up in the drainage water and ultimately accumulate in soil receiving drainage water and to monitor these chemical constituents. This approach has broad geographic application on marginally productive, arid-zone, irrigated agricultural lands throughout the world where water scarcity is a concern and the land is tile drained or has a water table sufficiently deep to allow adequate leaching. Another distinct contribution is an understanding of the consequences when drainage water reuse is terminated on a saline–sodic soil with high clay content (>40% clay) that has a water table 1.5 m or less from the soil surface. In less than 2 years the sodium and salinity levels returned to their initial condition and trace elements abruptly increased. So, even though the prognosis is favorable, reclamation may be temporary if downward water flow is not maintained since a shallow water table can become a source for salts and trace elements.

From 1999 to 2009, the application of 1.8–16.3 dS m⁻¹ drainage water on saline–sodic soil resulted in removal of salinity and Na from the top 1.2 m of the soil profile, especially from the top 0–0.6 m; removal of B from the top 0.6 m; and removal of Mo from the top 1.2 m; resulting in the reclamation of marginally productive soil and bringing it back into production. The evaluation of the long-term sustainability of drainage water reuse on marginally productive soil in California's WSJV has shown that drainage water reuse can be suitably managed (i) to improve the soil quality of a saline–sodic soil by leaching salinity, Na, and detrimental trace elements (B and Mo) thereby reclaiming the soil and returning it to productivity, (ii) to transform drainage water from an environmental burden into a water resource that produces forage to support livestock, and (iii) to reduce dramatically the volume of drainage water disposed in evaporation ponds thereby freeing up land that must be set aside for evaporation ponds and returning it to production.

From the perspective of impacts on soil chemical properties, there are few detrimental impacts of concern from drainage water reuse on saline–sodic soil. Even so, it is advisable to monitor certain soil chemical properties during drainage water reuse in the WSJV. An initial steady increase in pH_e in the first few years is ostensibly the only area of potential concern since elevated pH_e can influence macronutrient (*i.e.*, P) and micronutrient (*i.e.*, Mn, Fe, Cu, Zn, and Co) availability. However, pH_e decreased from 2002 to 2004 and leveled off from 2004 to 2009 even though the reused drainage water applied for irrigation consistently had pHs above 8.4; consequently, pH_e does not appear to be a significant concern, but is still worthy of monitoring during drainage water reuse. The accumulation of Se at the north and south ends of the field did not pose any threat, but continued monitoring is advisable. The recycling of Mo to the soil surface by sweet clover also does not threaten the viability of reusing drainage water, but should be monitored since

sufficiently high levels of Mo could accumulate in the vegetative portion of sweet clove to affect the digestive tract of ruminant livestock. The rapid return of the reclaimed soil to its original condition once irrigation terminated is not so much a concern as it is a matter needing awareness. Specifically, reclamation efforts can be quickly reversed in areas such as the WSJV, where a shallow water table serves as a sink for salts and trace elements when the net water flow is downward due to irrigation but serves as a source of salts and trace elements when the net flow is upward due to the depletion of water from the root zone by plants, which creates a potential gradient that draws water upward from the shallow water table. The upward flow of water brings with it dissolved salts and trace elements that had previously been pushed below the root zone when irrigation was occurring.

The San Joaquin Valley is unquestionably one of the world's most agriculturally productive regions due to a year-round arid climate that is conducive to high crop yields and a source of irrigation water that is of excellent quality. However, urban demands for water and repeated droughts have resulted in tremendous pressure placed on San Joaquin Valley producers to reduce their high consumptive water use, which has led to the serious consideration of alternative water sources. This study has clearly shown that drainage waters can serve as a viable alternative water supply and concomitantly reclaim saline–sodic soils; thereby, reducing drainage water volumes and decreasing the need for evaporation ponds while returning marginally productive land to production. Drainage water reuse on a saline–sodic soil is sustainable for a minimum of 10 years and is likely to be sustainable well beyond that timeframe, provided that attention is given to observed slow but steady increases in pH.

The ramifications of this study for the WSJV are noteworthy. In the WSJV approximately 340 000 ha are affected by shallow water tables. The land needed for evaporation ponds to handle the drainage water from this land is about 31 000 ha. The reuse of drainage water on 3000 ha of saline–sodic soil growing salt-tolerant crops would reduce the land needed for evaporation ponds by roughly 90%.

Demonstration of the sustainability of drainage water reuse in WSJV is of greatest significance with respect to its value to current joint research efforts by USDA and the US military to improve the competitiveness of biofuels with petroleum fuels by lowering feedstock-associated costs. The price of biofuel from plant oil is 10% conversion technology-related and 90% feedstock associated. A significant step toward the competitive commercialization of biofuel for private and military utilization is evidence of the sustainability of drainage water reuse on salt-tolerant crops in the WSJV. The US military desires a secure fuel source with prices unaffected by world events. Brassica oil seed crops have been identified as biofuel feedstocks adapted to western U.S. conditions that are potentially sustainable for producing commercial and military aviation fuels. Technology already exists for producing hydrotreated renewable jet (HRJ) fuel from agricultural sources of lipids such as oil seed crops. The west side of California's San Joaquin Valley (WSJV) is ideal for feedstock production since it contains substantial marginally productive land and has degraded waters that could be used to produce feedstocks without competing with higher valued crops and land uses. By growing salt-tolerant oil seed crops, such as

canola (*Brassica napus* L.) on marginally productive saline-sodic soils using low-cost degraded waters that are unsuited to the economical production of conventional crops, the feedstock-associated costs are reduced, helping to make biofuel more economically competitive. Combining these cost savings with superior performing germplasm and enhanced production methods can yield biofuels competitive with petroleum fuels. Therefore, a key component of redesigning the oil seed feedstock supply chains to be commercially competitive is the demonstration of the long-term sustainability of drainage water reuse established by this study.

The extensive spatio-temporal dataset of this study and demonstrated methodology for monitoring management-induced changes using EC_a-directed soil sampling are additional significant assets of this study. There are no other known spatial datasets recording the impact of drainage water reuse over such an extensive time period. Furthermore, the methodology for mapping and monitoring degraded water reuse impacts provides a means of assessment that assists the producer in site-specific management to identify where, when, and the amounts of irrigation or soil amendments that are needed to maintain the sustainability of degraded water reuse as well as providing the information that is needed for crop selection.

Abbreviations

EC _a	Apparent soil electrical conductivity (dS m ⁻¹)
EC _e	Electrical conductivity of the saturation extract (dS m ⁻¹)
EMI	Electromagnetic induction
EM _h	Electromagnetic induction measured in the horizontal dipole mode
EM _v	Electromagnetic induction measured in the vertical dipole mode
SAR	Sodium adsorption ratio
WSJV	West side of the San Joaquin Valley.

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